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Impact of structural changes in wood-using industries on net carbon emissions in Finland

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ABSTRACT

Forests and forest industries can contribute to climate change mitigation by sequestering carbon from the atmosphere, by storing it in biomass and by fabricating products that substitute more greenhouse gas emission intensive materials and energy. The objectives of the study are to specify alternative scenarios for the diversification of wood products markets and to determine how an increasingly diversified market structure could impact the net carbon emissions of forestry in Finland. The net carbon emissions of the Finnish forest sector were modelled for the period 2016–2056 using a forest management simulation and optimization model for the standing forests and soil, and separate models for product carbon storage and substitution impacts. The annual harvest was fixed at approximately 70 Mm^3, which was close to the level of roundwood removals for industry and energy in 2016. The results show that the substitution benefits for a reference scenario with the 2016 market structure account for 9.6 Mt C (35.2 Mt CO_2 equivalent [CO_2eq]) in 2056, which could be further increased by 7.1 Mt C (26 Mt CO_2eq) by altering the market structure. As a key outcome, increasing the use of by-products for textiles and wood-plastic composites in place of kraft pulp and biofuel implies greater overall substitution credits compared to increasing the level of log harvest for construction.

Keywords: bioeconomics; carbon emissions; forest products; industrial ecology; national forest inventory; substitution

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INTRODUCTION

One of the core targets of the Paris Agreement is "*to achieve a balance between anthropogenic emissions by sources and removals by sinks of greenhouse gases in the second half of this century*" (UN 2015). The agreement also recognises the role of maintaining and managing forests towards this objective. On one hand, forests contribute to climate change mitigation by sequestering carbon from the atmosphere to trees and soil (Nabuurs et al. 2017). On the other hand, using harvested wood products in place of more fossil emission intensive materials and energy sources can yield substitution benefits in other economic sectors (Sathre and O'Connor 2010; Leskinen et al. 2018). It depends on the scale of the substitution benefits and the carbon stocks of wood-based products, whether a trade-off exists in the short-term between increasing wood harvesting and maintaining a larger net carbon sink.

The global forest sector has experienced major changes in the 2000s (Hansen et al. 2013), and is expected to do so on a pronounced scale in the future, partly driven by circular bioeconomy ambitions (Hetemäki et al. 2017). Notably, graphic paper industries have been dwindling due to substitution by electronic media, which has motivated forest industries to rethink their product portfolios (Näyhä and Pesonen 2014; Hetemäki and Hurmekoski 2016). As a consequence, the European forest product markets are no longer likely to follow the trends of the 20th century (Jonsson et al. 2017). To replace some of the declining businesses, the wood using industries are expected to seek growth at least in the construction, textile, chemical (including polymers), and liquid biofuel markets (Hurmekoski et al. 2018). This could have important consequences for the overall GHG substitution benefit of wood utilisation, as wood may increasingly replace other materials and energy carriers in the emerging markets. Simultaneously, the use of wood is expected to decrease for graphic paper production, to which no significant substitution benefit can be attributed (Achachlouei and Moberg 2015).

Assessing the net carbon emissions of wood use requires that the carbon fluxes in both the forest ecosystem and related technosystem are quantified. The forest ecosystem emits CO_2 from respiration, removes CO_2 from the atmosphere by photosynthesis, and stores carbon in above- and below-ground biomass and soil. Harvested wood biomass is transferred to the technosystem, which stores carbon in wood products, emits CO_2 from biomass combustion and fossil fuel based operations and avoids fossil emissions due to material and energy substitution. In addition to CO_2 emissions, the forest ecosystem and technosystem cause other greenhouse gas (GHG) emissions such as methane (CH_4) and nitrous oxide (N_2O), and their contribution to climate change impacts can be meaningful. The climate impact assessment of wood utilization can be simplified by converting all GHG emissions to CO_2 equivalents on a 100-year time horizon. However, forests may have several other climate impacts than GHG emissions, such as albedo, biogenic aerosols, effect on cloud formation, evaporation and surface roughness (Kalliokoski et al. 2019).

In Finland, the carbon sequestration and stocks of boreal forests have both increased substantially since the 1970's (Finnish Forest Research Institute 2014). This is especially because the mean annual volume growth has clearly been greater than the mean annual amount of wood harvested (e.g. about 100 and 60 Mm^3 yr^{-1}, respectively, in 2004-2013). Climate change and intensified forest management have also contributed to this development (Hynynen et al. 2015; Henttonen et al. 2017; Finnish Forest Research Institute 2014). Large uncertainties exist in the projected climate change (Ruosteenoja et al. 2016), which together with prevailing environmental conditions, forest structure, disturbance regime, and forest management intensity, will likely affect the development of forests in different boreal regions (Kellomäki et al. 2018; Heinonen et al. 2018a, 2018b).

The climate impacts of the forest-based bioeconomy have been assessed in several studies by combining the carbon fluxes of the forest ecosystem and the technosystem. According to Heinonen et al. (2017), the largest sustainable amount of domestic logging of sawlogs and pulpwood in Finland, without endangering the stable wood supply, is 73 million $m^3\ yr^{-1}$ for the next 90-year period (not considering the impacts of climate change). The total carbon balance of forestry was found to be the highest with lower volumes of roundwood removal. On the other hand, with intensified forest management (forest fertilisation, improved regeneration material and ditch network maintenance), the sustainable harvest level could be increased by about 4 million $m^3\ yr^{-1}$ under a changing climate over a period of 90 years (Heinonen et al. 2018a). Intensified management under moderate (RCP4.5) climate change would make it possible to harvest about 80 million $m^3\ yr^{-1}$, without decreasing the growing stock volume, over the next 90 years (Heinonen et al. 2018b).

Baul et al. (2017) calculated the net climate impacts for year 2016-2055 in southern Finland using the currently recommended forest management scenario as a reference. In their study, maintaining higher stocking with earlier final felling and the use of logging residues as energy appeared to be the best option for increasing both the climate benefits and economic returns. On the other hand, the use of alternative assumptions concerning GHG displacement potential largely affected the mitigation potential of forest biomass. Soimakallio et al. (2016) found that an increased harvesting and use of wood may not result in climate benefits for 100 years, due to the decline in the forest carbon sink. Thus, there appears to be a significant trade-off between avoiding emissions through fossil fuel substitution and reducing the forest carbon sink due to wood harvesting, despite a significant compensation from substitution impacts. Seppälä et al. (2019) calculated a *required displacement factor* for Finland, suggesting that in order to achieve net carbon emission reduction with increased harvest, the average displacement factor of wood use should be between 2.0 and 2.4 tC/tC.

Previous studies have rarely analysed in depth how the changes in the market structure would influence the substitution patterns and the net carbon emissions. The ambition of this study is to highlight the impact of changing market structures and consequent substitution patterns, as this yields a more realistic picture of the possible contribution of the substitution effect on the net carbon emissions. Towards this backdrop, the objectives of the study are to i) define alternative market structures for the future use of wood resources in Finland, based on the expected decline of graphic paper markets and the growth of emerging wood-based products, ii) define displacement factors for established and new wood-based products, weighted by the end uses of intermediate products; and iii) based on these, determine how an increasingly diversified market structure may impact the net carbon emissions of forestry in Finland.

MATERIALS AND METHODS

Computing the net carbon emissions for a forest-based production system

The net carbon emissions (NCEs, in CO_2 equivalents) for a forest-based production system include the carbon fluxes in the forest ecosystem and the related technosystem (Figure 1).

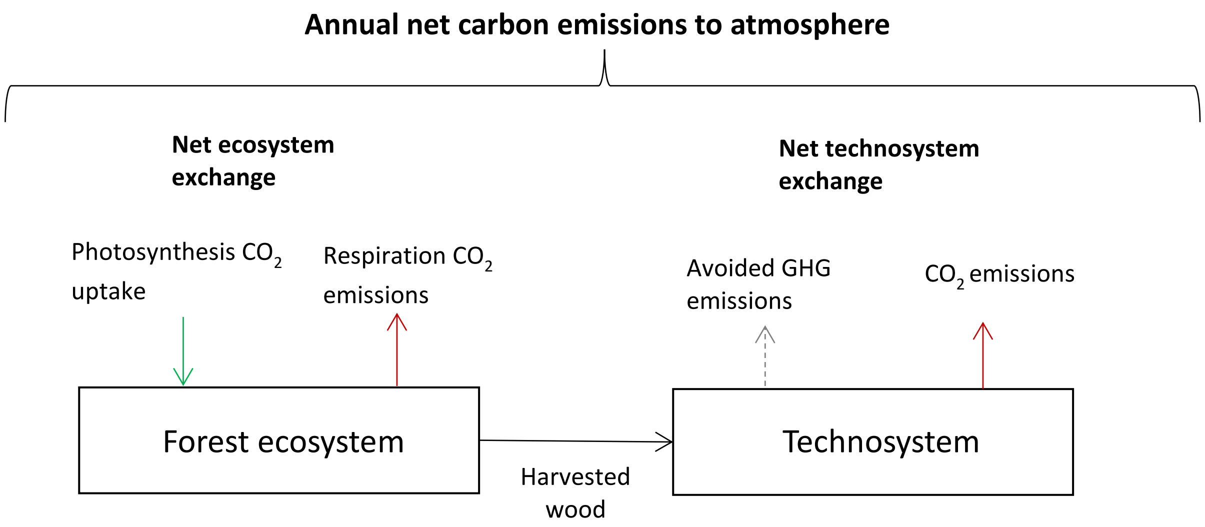

Fig. 1. Schematic figure of annual net carbon emissions to the atmosphere caused by carbon flows in a forest ecosystem and technosystem (Gustavsson et al. 2017).

The NCEs were calculated according to Equation 1:

$$NCE_t = (TC_{t-1} - TC_t) + (SC_{t-1} - SC_t) + (PC_{t-1} - PC_t) - SUB\_P_t - SUB\_EOL_t \qquad \text{(Eq. 1)}$$

, where TC = tree carbon stock, SC = soil carbon stock, PC = product carbon stock, SUB_P_t = avoided carbon emissions due to the substitution impact of harvested wood products (HWPs) at the production stage, SUB_EOL_t = avoided carbon emissions due to the substitution impact of HWPs at the end-of-life stage (utilisation as energy), and t = year.

The NCEs are reported as annual changes in the carbon stocks and as annual substitution impacts. A positive value of NCE translates to an increase in the net carbon emission to the atmosphere, while a negative value translates to the opposite.

Ecosystem carbon flows and assumptions

Calculation of carbon flows in forests

We calculated the carbon flows in forests (living forest biomass and soil) using Monsu software (Pukkala 2011), which has been used in several scenario analyses of the impacts of forest management and harvesting intensity on the timber supply and carbon balance of forestry (Heinonen et al. 2018a, 2017, 2018b; Pukkala 2011, 2014, 2017; Zubizarreta-Gerendiain et al. 2016). The Monsu software is able to calculate changes in all the carbon pools listed in the Intergovernmental Panel on Climate Change's (IPCC´s) carbon accounting rules, including: (1) living forest biomass; (2) soil organic matter; and (3) wood-based products and fuels. In this study, however, the third component was modelled separately.

The initial carbon pool for soil organic matter was calculated with models (Pukkala 2014) at the start of the simulation. Tree species-specific turnover rates were used to calculate the litter production from tree biomass (Pukkala 2014). The inputs to the soil carbon stock included dead organic matter from litter, mortality, harvest residues (including tree tops, roots, branches, needles/leaves and bark) and the growth of peat in undrained peatland forests. The release of carbon through the decomposition of dead organic matter (CF_b) was simulated using the Yasso07 model (Tuomi et al. 2011b; Liski et al. 2009; Tuomi et al. 2011a).

Forest data and the simulation of treatment schedules

We used a sub-sample of the sample plots of the 11th National Forest Inventory (NFI11, 2009-2013) of Finland (Korhonen 2016) as input data for the simulations. The forest data included one sample plot from every inventory cluster. The plots were located on forestland assigned to timber production. A total of 1890, 1393 and 1402 plots were used for southern, central and northern Finland, respectively (Heinonen et al. 2018a, 2017, 2018b). Country-level results were aggregated from the separate analyses performed for these three regions.

Different treatment schedules were simulated for each sample plot for ten 10-year periods, resulting in a100-year simulation period (Table 1). A sample plot was managed with a particular treatment if the predefined conditions for such treatment were fulfilled in the middle of a 10-year period. In the simulations, the growth responses to climate change of different boreal tree species were considered by employing a new meta model approach (Seppälä et al. 2019). The model is based on the measured growth responses of Scots pine trees in provenance trials in Finland and Sweden (Beuker 1994; Persson and Beuker 1997; Berlin et al. 2016) and the growth responses predicted by an ecosystem model (Kellomäki et al. 2008, 2018). In the simulations, we assumed a mild climate change – the RCP2.6 forcing scenario of the Coupled Model Intercomparison Project 61 Phase 5 (CMIP5). In this scenario, the mean annual temperature is likely to increase by 2°C and annual mean precipitation by 6% in Finland, if the atmospheric CO_2 concentration increases to 430 ppm by 2100 (Ruosteenoja et al. 2016).

Table 1. Simulated management activities at different sites.

Forest regeneration	If the amount of natural advance regeneration was insufficient, sub-xeric sites were seeded with Scots pine, and mesic sites were planted either with Scots pine, Norway spruce or silver birch (probability 0.3, 0.6, 0.1), and herb-rich (or better) sites by Norway spruce or silver birch (probability 0.8 and 0.2). Other upland forest sites and all drained peatland sites were regenerated naturally. Seedlings were also expected to be born naturally at all sites, regardless of regeneration method. Sown seeds and planted seedlings had 10% higher growth rates than naturally born seedlings (Heinonen et al. 2018a, 2018b). Necessary regeneration treatments were performed immediately after clear cutting.
Tending treatments in seedling stands	Tending treatments for dense seedling and sapling stands were simulated according to Finnish management recommendations (Äijälä et al. 2014).

Thinning and final cutting	The earliest possible time for thinning (uniform) and final felling (clear cutting) were defined by multiplying currently recommended (Äijälä et al. 2014) lower threshold values for thinning (basal area) and final felling (diameter at breast height) by 0.9. Also treatment schedules with postponed cuttings and a schedule without cuttings were simulated (Heinonen et al. 2018a, 2018b).
Fertilisation	25% of those upland sub-xeric pine-dominated and mesic spruce-dominated stands were fertilised where the growing stock characteristics fulfilled a set of pre-determined criteria (Heinonen et al. 2018a, 2018b). Fertilisation was not simulated simultaneously (same 10-year period) with a thinning treatment.
Ditch network maintenance	On 40% of the drained peatlands, the ditch network was re-excavated to maintain the ditches when the growing stock characteristics fulfilled a set of pre-determined criteria (Heinonen et al. 2018a, 2018b).

Harvesting scenarios and optimisation

In all wood-harvesting scenarios, the annual amount of harvested roundwood was fixed at 70 Mm^3. This roughly corresponds to the realised annual harvest of sawlog and pulpwood in Finland in 2016 (Natural Resources Institute of Finland 2017). In the reference scenario, the regional cutting targets for sawlog and pulpwood during the first five 10-year periods were derived for different tree species from the policy scenario in Lehtonen et al. (2016) separately for the three regions. The harvest targets of the last five periods were the same as during the fifth period.

The objective of the treatment scheduling problem was to maximise the timber production and profitability of forest management (net present value with a 3% discount rate), with species-specific even-flow harvesting targets for sawlog and pulpwood in each 10-year simulation period. More importance was given to fulfilling the harvesting targets of the earlier 10-year periods, if the cutting target could not be met during every period. The simulation and optimisation methods used in this study have been described in detail in Heinonen et al. (2018a, 2017, 2018b).

Technosystem carbon flows and assumptions

Scenarios

Eight scenarios depicting plausible changes in the technosystem were defined (Table 2). All scenarios were based on the same annual harvest level (70 Mm^3 yr^{-1}); that is, the scenarios only differed in terms of market structure. The exact assumptions on market structures are described in supporting information 1.

In Scenario 2 (Scenario 3), log production was increased (decreased) by 10% and pulpwood production was decreased or increased by an equivalent wood amount to keep the same annual harvest level. This was done because producing more sawlogs was expected to positively influence the net carbon emissions (Pingoud et al. 2010), and because literature has emphasized the role of wood construction in climate change mitigation (Smyth et al. 2017). These changes in sawlog and pulpwood proportions were modelled by varying the top diameter threshold of sawlogs and by altering the harvesting targets of sawlogs and pulpwood.

Table 2. Scenarios simulated in this study.

Scenario	**Description**
1. Reference	BaU (adopted from Lehtonen et al. (2016)) – Annual harvest in 2050 is 70 Mm^3
2. Sawlog +10	BaU, except for increased sawlog supply by 10% and decreased pulpwood supply by an equivalent share
3. Sawlog -10	BaU, except for decreased sawlog wood supply by 10% and increased pulpwood supply by an equivalent share
4. Biorefinery	Liquid biofuels and biochemicals account for 50% of all by-product use
5. Textiles	In addition to scenario 4, 50% of pulp production goes to dissolving pulp and further mostly to textiles
6. Composites	50% of pulp production is dissolving pulp (textiles); chemicals account for 50% of pulp side streams; composites and wood-based panels account for 50% of sawn wood residues
7. Graphic papers	BaU, except for assuming a negative DF (-0.58) for graphic paper production
8. Decarbonisation	BaU, except for assuming the average emissions of the energy sector to reduce by 80% by 2056 and consequently change the substitution impacts

Scenarios 4 to 6 were motivated by literature on emerging wood products, and the consequent reallocation of wood flows in terms of by-product utilization (Hurmekoski et al. 2018; Kunttu et al. 2019). Currently, most sawmilling by-products are utilised in mill energy production, district heating and electricity generation, while the rest is used for pulping. With the emerging renewable energy technologies, increasing nuclear power, and improving energy efficiency, the pulping by-products were assumed to be increasingly used for emerging wood-based products (Scenarios 4–6). Additionally, in scenarios 4-6, communication paper production was assumed to decline towards 2050, due to digital media replacing print media. Similarly, the production of mechanical pulp was assumed to have terminated by 2050, with Norway spruce pulpwood being used for kraft and dissolving pulp instead.

Emerging biorefineries are expected to produce bioethanol and renewable diesel, and platform chemicals to be refined to polymers and various other chemicals (Scenario 4). In terms of volume, we assumed the chemicals to be drop-in or smart drop-in chemicals[1], for use as direct substitutes for fossil-based ones. Dornburg et al. (2008) identified ethylene, used mostly for polyethylene (PE), as being the most important bio-based intermediate chemical. It is also the largest of the currently produced petrochemicals, by volume.

The production of dissolving pulp was assumed to grow significantly, driven by the increasing demand for man-made cellulosic fibres for textiles (Scenarios 5 and 6) (Pöyry Inc. 2015). In addition, in Scenario 6, a major shift from the use of by-products as energy to long-lived wood products (wood-plastic composites and wood-based panels) was assumed (Kunttu et al. 2019).

[1] Bio-based drop-in chemicals such as ethylene are chemically identical to existing fossil-based chemicals. Smart drop-in chemicals refer to a special subgroup of drop-in chemicals such as succinic acid whose bio-based pathways provide advantages compared with the conventional petrochemical pathways, notably a comparably high biomass utilization efficiency, low embodied energy, noncomplex pathway, or low toxicity (Carus et al. 2017).

In addition to the market scenarios (1–6), two additional scenarios (7 and 8) were formulated to test the sensitivity of the substitution impacts to some of the most important assumptions. Firstly, graphic papers form one of the largest forest product categories by volume, yet only one single study could be found that quantified the possible substitution impact between graphic paper and electronic media (Achachlouei and Moberg 2015). To consider this uncertainty, we included one scenario assuming a negative substitution impact for graphic papers (scenario 7).

Secondly, the emissions of the energy sector need to be reduced significantly towards 2050, to meet the ambitious targets of the Paris Agreement. This is likely to reduce the relative climate benefit of wood products (Peñaloza et al. 2018), as the emissions of wood-based products cannot be assumed to decline at a similar pace as for the alternative products, because the fossil-based energy input for the production of wood-based products in Finland is already relatively low. In an attempt to explore the consequences of this major change, we estimated the impacts of decreasing the GHG emissions from energy production for the displacement factors of production and end-of-life stages (Scenario 8).

Displacement factors

The displacement factor, DF_i, for wood product *i* was calculated as:

$$DF_i = \frac{GHG_{alternative} - GHG_{wood}}{WU_{wood} - WU_{alternative}} \qquad \text{(Eq. 2)}$$

, where $GHG_{alternative}$ and GHG_{wood} are the GHG emissions resulting from the use of the non-wood and the wood alternatives expressed in mass units of carbon (C) corresponding to the CO_2 equivalent of the emissions, and WU_{wood} and $WU_{alternative}$ are the amounts of wood used in the wood and non-wood alternatives, expressed in mass units of C contained in the wood (Sathre and O'Connor 2010).

The GHG emissions in Eq. 2 were expressed as CO_2 equivalents in a 100-year time frame. The GHG emissions were converted to a unit of carbon (C) by multiplying CO_2 equivalent (CO_2eq) by 12/44. Also, the amounts of wood used were converted to carbon, resulting in a unit of tC/tC for the DF. Here, we assumed a density of 460 kg/m^3 for Scots pine, 410 kg/m^3 for Norway spruce, and 580 kg/m^3 for birch (Repola 2009). The carbon content of wood was assumed to be 50%. In cases where DF was determined per total wood use, we used roundwood equivalent (RWE) conversion factors by UNECE/FAO (2010) and Hurmekoski et al. (2018).

The DFs were determined mainly using publicly available domestic data sources, but in several cases also other data sources and Ecoinvent database were utilized. Data for sawnwood and plywood DFs were obtained from Vares and Häkkinen (2017), in which functionally similar wooden construction elements' climate change impacts were compared to construction elements based on steel and concrete. DFs were determined considering the mass of different construction elements used in a representative building.

The DF for packaging was obtained from Knauf et al (2015). Packaging (container boards, carton boards and sack paper) were assumed to replace plastic, glass and metal packaging (Knauf et al. 2015). Mechanical pulp was assumed to be used for magazine paper and no substitution credit was given. However, the DF for graphic papers depends on a wide range of factors such as the brightness of the screen of a mobile device and the number of readers, with one study suggesting that there could be an average DF of -0.58, instead of the zero (Achachlouei and Moberg 2015). This value was used for scenario 7.

DFs of diesel and ethanol were adapted from Bright et al. (2010) and Soimakallio et al. (2009). Ethylene was assumed to ultimately replace plastics in packaging. Viscose, derived from dissolving pulp, was assumed to replace cotton. The DF for textile replacement was adapted from the Ecoinvent database 3.0 (see also supporting information 2). Wood-plastic composites (WPC) were assumed to replace virgin polypropylene (PP) especially in car manufacturing (Judl et al. (2016), Ecoinvent database 3.0).

Combined heat and power (CHP) was assumed to replace fossil energy. Wood biomass, such as chips, residues and sawdust, are used typically mixed with other raw materials, especially peat. The production of pulp consumes large amounts of energy, the majority of which is produced using the by-products of pulp mills. As the generated energy was assumed to be used by the mills on site, no substitution credit was assumed for mill energy.

For the end-of-life DFs, we assumed that all wood material products would be used for energy at the end of their lifespan. This is currently a common practice in Finland and in most of the countries where Finnish wood products and intermediate products are exported, as landfilling of organic and biodegradable waste streams is forbidden. A widely used DF for energy displacement is 0.8 tCt/C (Pukkala 2014). In this study we assumed that by 2016 energy displacement would be slightly smaller due to the share of fossil energy having declined and the share of renewable energy increased in recent years. Thus, the DF used for the EoL energy displacement was 0.7 tC/tC. For viscose, the end of life energy DF was lower (0.23 tC/tC) as substituted cotton could be used as energy as well, but the energy efficiency was considered lower compared to viscose. No EoL substitution credit was assigned for chemicals.

In scenario 8, we assessed how the DFs and subsequent substitution impacts would change driven by the decarbonization of the energy sector. Decarbonization influences both product displacement and end of life energy displacement. We assumed that the GHG emissions of energy production would be reduced to 20% of the level of emissions in 2016 by 2056, and re-estimated the DFs accordingly. In line with this assumption, we assumed that only 20% of replaced energy would be fossil-based in 2056. Because of decarbonization, GHG emissions of processing of both wood products and their substitutes would decrease in the future. Assumed decarbonization decreases the carbon footprints of energy intensive products in particular. In order to determine new DFs because of decarbonization, we determined the proportion of GHG emissions caused by energy use for all substituted and wood-based products. Information on the share of energy production of the total climate change impacts of plastics, steel and concrete were obtained from Material Economics Sverige AB (2018) and for other products from Mattila et al. (2018). A linear decline in both the production stage and EoL DFs was assumed, with the EoL DF declining from 0.7 tC/tC in 2016 to 0.14 tC/tC in 2056 (see Table 3).

In addition to the two extra scenarios (7 & 8) highlighting the uncertainties in DF estimates, we carried out further sensitivity analysis for the DFs of textiles and WBCs by increasing and decreasing their DF values by 50%. This was done, due to the reliability of the estimates remaining relatively weak, and because they are important determinants of the overall net carbon emissions.

Substitution

The wood use in Eq. 3 refers to the amount of wood embodied in the end product, as the wood processing by-product flows were assigned to other markets, which were altered across scenarios. The alternative product and wood product were functionally equivalent for each DF.

The substitution impact of the production stage for HWP (SUB_P, Equation 4) was calculated with the help of DF_i and the production volume of wood product i (PV_i):

$$SUB\_P(t) = \sum DF\_P_i(t) \cdot PV_i(t) \qquad \text{(Eq. 4)}$$

, where DF_P_i is the avoided fossil-based carbon emissions (tC) per carbon contained in product i used (tC) in the production stage. This also includes substitution effects caused by the upstream emissions of products.

In addition to material substitution, wood products can be used for energy substitution purposes in their end-of-life stage. We assumed that wood products, for which the only possible end-of-life use is combustion, would replace fossil fuels and merit a substitution credit (Table 3). The end-of-life substitution impact was calculated according to Equation 5:

$$SUB\_EOL(t) = \sum DF\_EOL_i(t) \times Outflow_i(t) \qquad \text{(Eq. 5)}$$

, where DF_EOL_i is the avoided fossil-based carbon emissions (tC) per carbon contained in product i used (tC) for the end-of-life stage (incineration), and $Outflow_i$ is the outflow of wood product i from the HWP pool (MtC yr^{-1}).

The total substitution impact of HWP is the sum of *SUB_P* and *SUB_EOL*. The average production stage displacement factors were calculated by summing up the production stage substitution impacts of all products and dividing the sum by the amount of carbon contained in total harvest (see also Figure 2, Table 3, and Supporting Information 3). The average end-of-life displacement factors were determined by dividing the end-of-life substitution impacts of all discarded wood products by the amount of carbon contained in the outflow from the HWP pool. We reported the average displacement factors for the production stage and the end-of-life stage separately, as the divisors are not necessarily comparable, in that one originates from annual harvest and the other from historical harvests as the HWP pool gradually decays.

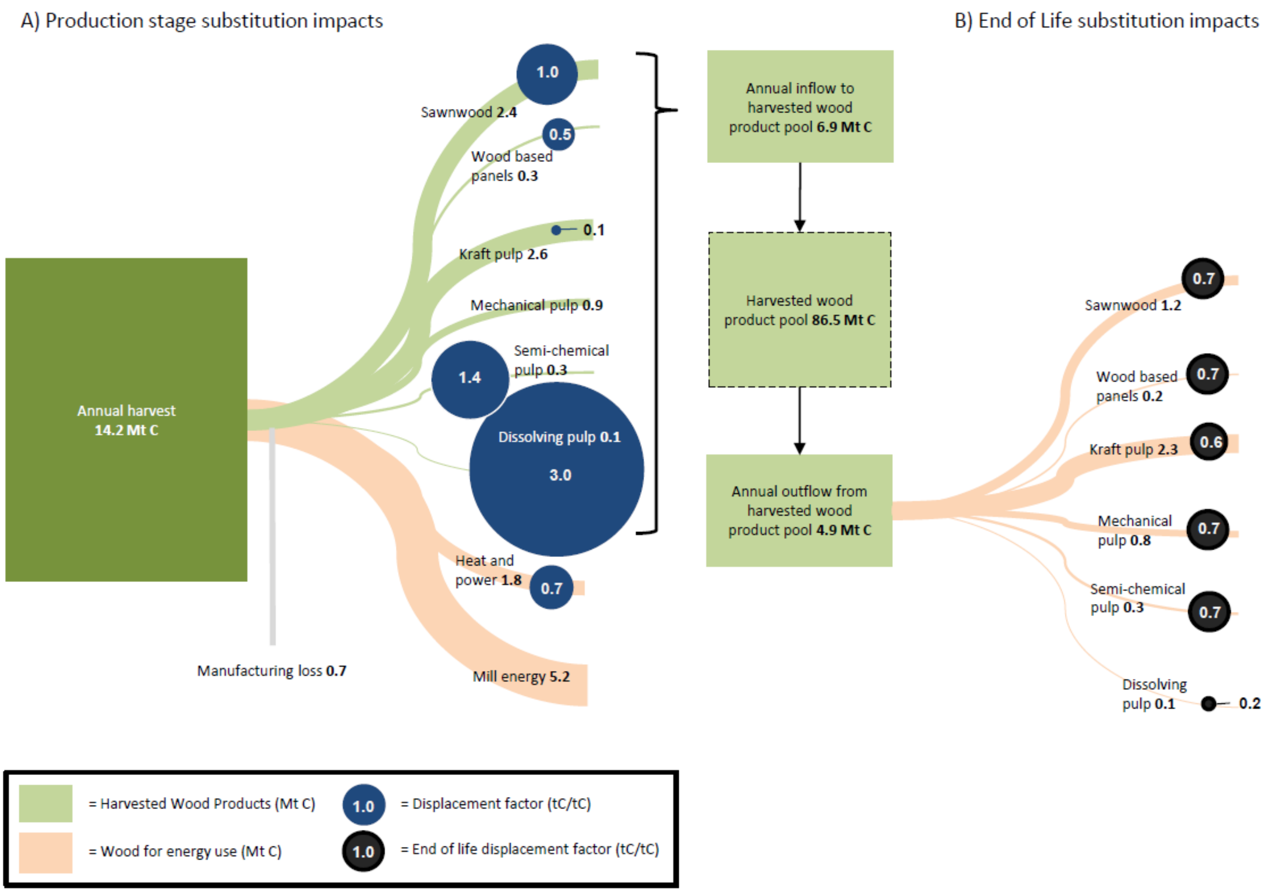

Figure 2. Wood use allocation and respective weighted displacement factors estimated for 2016. Part (A) shows the amount of carbon (Mt C) in the wood used for each intermediate wood product and the respective weighted displacement factors. Part (B) shows the amount of carbon (Mt C) in discarded

wood products (outflow from the HWP pool) and the respective weighted end-of-life displacement factors. The combined substitution impacts of (A) and (B), calculated as the product of wood flows and weighted displacement factors, yields the total substitution impact in a given year.

The harvests were allocated for intermediate products that could be used for a range of end products. To determine a weighted DF for each intermediate product, the end-use distribution of each intermediate product was specified (Table 3). As no single data source was available, the end-use shares were based on various sources (Poyry Inc 2017), and required assumptions particularly when it came to the use of different pulp varieties for certain paper grades (Pöyry Inc. 2015). The level of detail in the assumptions was limited by the complexity of the market and the consequent lack of data. For example, due to a lack of more detailed or consistent data, we had to assume the same DFs for the sawnwood produced from pine and spruce, despite the possibly differing end uses.

Product carbon storage

The annual change in product carbon storage was calculated according to the IPCC's general model called 'Product in Use' (Pingoud et al. 2006) in which estimates of the change in carbon are made by tracking carbon inflows to, and outflows from, the 'products in use' carbon pool. The amounts of inflow and outflow are influenced by product half-life periods (see Table 3). A product half-life is the number of years it takes for one half of the material from the product stock to return into the atmosphere. The carbon inflow to the pool was estimated by using the production volumes of different HWP scenarios. The carbon pool was calculated from 1955 to 2056. The wood flows to 2016 were estimated based on annual harvesting levels, and assuming the same production structure as in 2016.

We assumed a 5% manufacturing loss for all other products except for ethanol and ethylene. For these two products, the manufacturing loss was 88% and 93%, respectively, due to their very low biomass utilisation efficiency (Iffland et al. 2015). Some components could, in principle, be separated, and used to produce co-products, but due to a lack of robust data, and the possible variability between biorefineries, these possibilities were not considered. We assumed no direct roundwood flow to incineration; that is, only by-products could be used as bioenergy. Also logging residues were not within the system boundary of the used wood product model, i.e., no substitution credit or wood use were assigned to them.

Additional sensitivity analysis was carried out also for the calculation of the harvested wood product stock changes by reducing the HWP half-life coefficients by 50%. This would correspond, for example, to the estimate of Pingoud et al. (2001) sating that the average lifetime of buildings in Finland is less than 40 years. That is, earlier studies strongly suggest that the IPCC default coefficients are systematic overestimates, as there are substantial losses in wood material both in the construction phase and in the recovery of dismantled wood products for energy use, and because a higher share of wood products may end up in short-term uses or need to be replaced sooner than estimated (Statistics Finland 2010).

Table 3. End uses, displacement factors, and product half-life values of intermediate products.

Intermediate product	End use share	Displacement factor (DF_P) (tC/ tC) 2015	Displacement factor (DF_P) (tC/ tC) 2056 (scenario 8)	Half-life (years)
Solid wood products				
Sawnwood (incl. EWPs)	69 % construction	1.10	0.82	35
	19 % packaging (pallets)	1.10	-0.02	35
	3 % furniture	0.90	0.12	35
	9 % other	0	0	35
Plywood	41 % construction	1.10	0.74	35

	59 % other	0	0	35
Pulp and paper				
Mechanical pulp	100 % graphic papers (incl. newsprint)	0	0	2
Semi-chemical pulp	100 % packaging (container boards)	1.40	0.30	2
Kraft pulp	38 % graphic papers	0	0	2
	45 % hygienic: tissue, toilet, etc. (incl. fluff pulp)	0	0	2
	8 % packaging (carton boards, sack paper)	1.40	0.30	2
Dissolving pulp	74 % Textiles (viscose)	4.00	2.00	2
	26 % other	0	0	2
Heat and power				
CHP	100 % energy	0.70[1]	0.14	1
mill energy	100 % energy	0[2]	0	1
Biorefining				
diesel	100 % transport fuel	0.63	N.A.[3]	2
ethanol	100 % transport fuel	0.70	N.A.	2
ethylene	100 % packaging (PE, PET)	1.3	N.A.	2
Composites				
wood-plastic composite	70 % terraces	0	0	
	30 % plastic components for cars (polypropylene)	7.38	3.20	35

[1] assuming fossil energy substitution
[2] no substitution credits for mill energy has been given as the energy is used for fuelling the production process of wood-based products.
[3] N.A. = Not applicable (data not used)

RESULTS

The scenarios altering the ratio of sawlog and pulpwood supply had a minor impact on carbon stock changes in forests, soil and products (Fig. 3). The only exception was the standing tree biomass in the period of 2046-2056. This can be explained by a temporary deceleration of tree growth when targeting an increased harvest of sawlogs. The peak in soil carbon at the beginning of the simulation period was likely due to an immediate intensification of young forest management, which caused a surge of rapidly decaying litter to the soil.

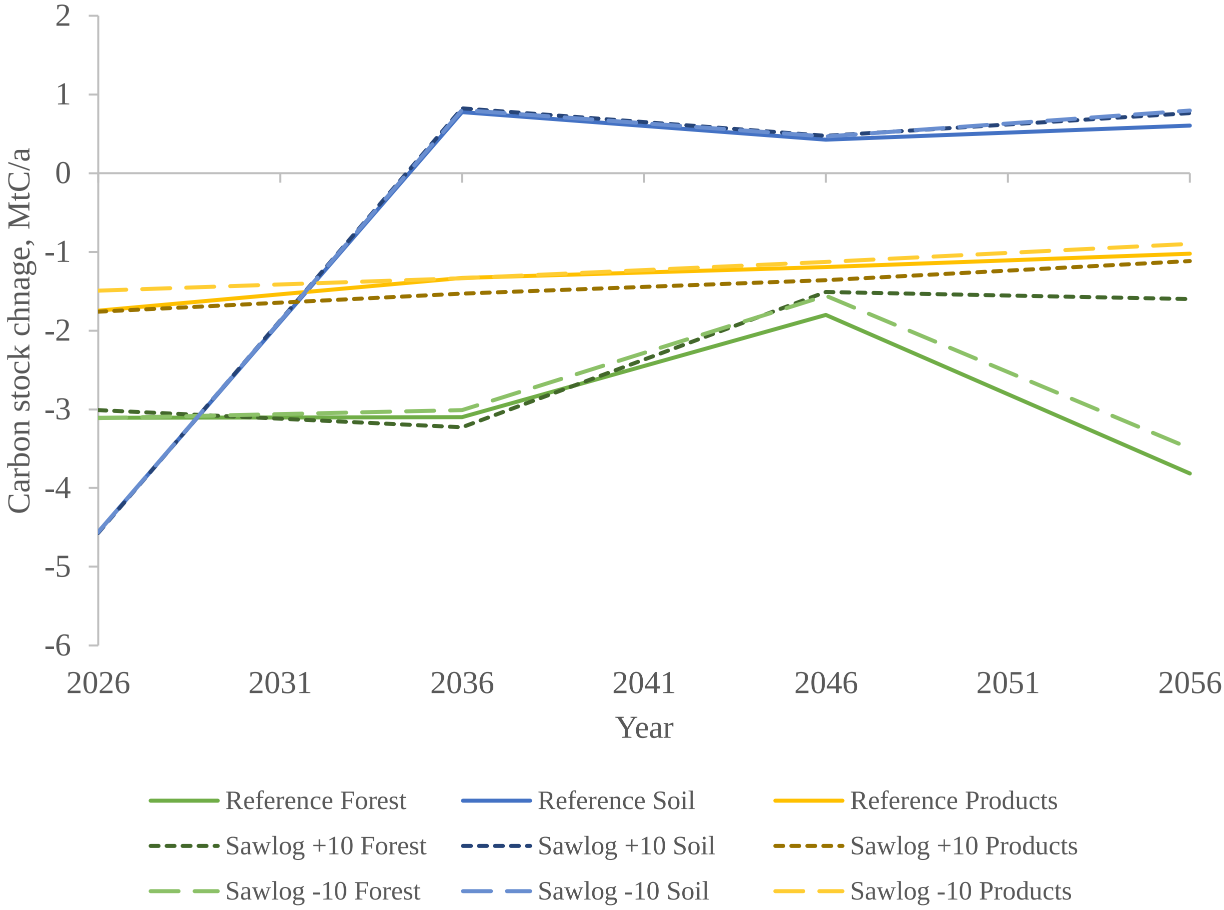

Figure 3. Carbon stock change in forest, soil and products in scenarios 1–3. Underlying data used to create this figure can be found in the supporting information 4.

The scenarios showed vast differences in the substitution impact (Table 4). The average production stage displacement factors in 2056 ranged from a minimum of 0.3 tC/tC (Scenario 3 – log decrease and scenario 4 - biorefinery) to a maximum of 0.8 tC/tC (Scenario 6 - composites). The scenarios showed that altering the ratio of sawlog and pulpwood production had a relatively minor impact (Scenarios 2-3) compared to altering the end-use of pulp from graphic papers to textiles (Scenario 5) and the use of by-products from energy to long-lived products (scenario 6). This was partly due to a relatively minor alteration in the sawlog and pulpwood ratio which significantly limited the increase in the production of solid wood products. The biorefinery scenario (4) showed slightly lower substitution benefits compared to the baseline, which was mainly due to the extremely low efficiency of feedstock conversion to ethanol and ethylene, and the assumed lack of end-of-life substitution gains when compared to many paper products.

Table 4. Substitution impact (production and end-of-life) and average displacement factors at the end of the simulation period (in 2056).

	Total substitution, MtC	Average DF - production	Average DF - EoL
1. Reference	-9.6	0.33	0.63
2. Sawlog +10	-10.1	0.35	0.64
3. Sawlog -10	-9.2	0.30	0.63
4. Biorefinery	-8.7	0.30	0.36
5. Textiles	-13.0	0.60	0.30
6. Composites	-16.8	0.80	0.35
7. Graphic paper	-8.4	0.25	0.63
8. Decarbonisation	-3.0	0.14	0.13

With all of the carbon flows considered, the Finnish forestry sector remained a net carbon sink in all scenarios (Fig. 4). The harvest rate of 70 Mm^3 yr^{-1} first reduces the net sink, but in some scenarios this is compensated for by an increase in forest growth, or in substitution impacts. The difference in the net carbon emissions between the reference scenario and Scenario 6 (composites) in 2056 equalled 8.1 MtC. In comparison, the difference between the net carbon emission with and without considering the substitution benefits for the reference scenario was around 9.5 MtC. The substitution impact (including the production and end-of-life stages) constituted 42–84% of the total net carbon emissions. This was partly a result of a stable harvest level over the projection period that was close to the maximum sustainable harvest.

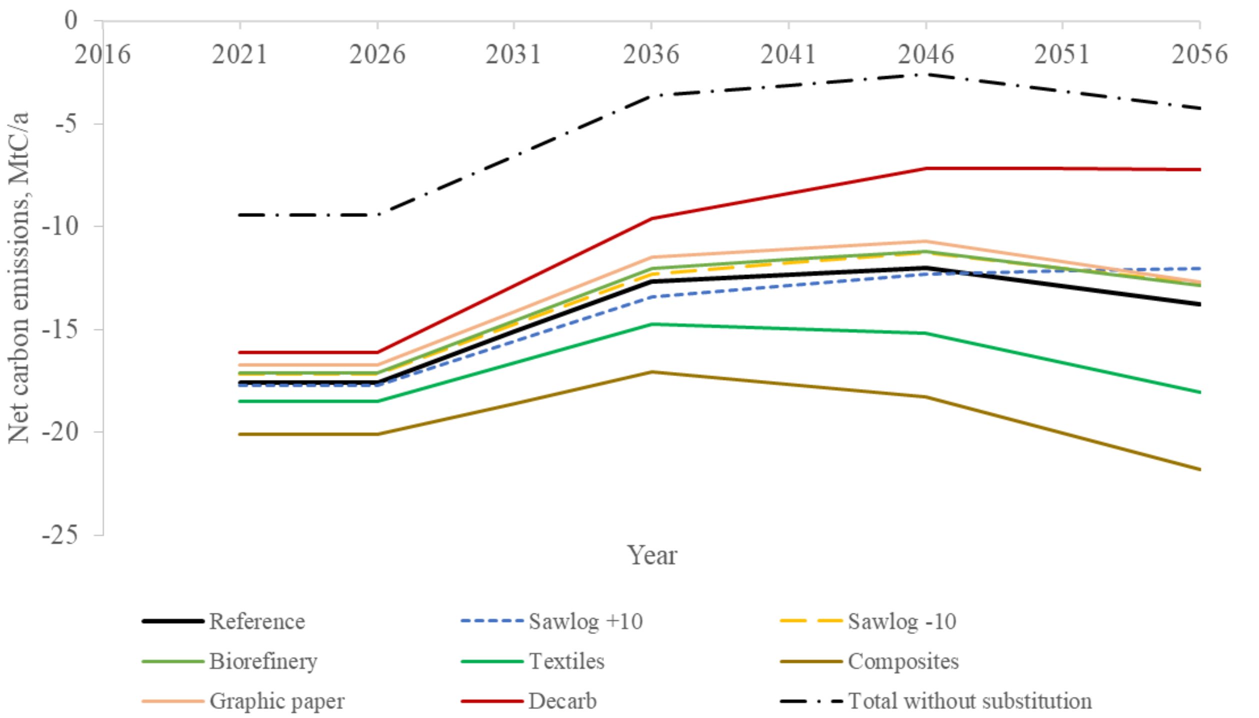

Figure 4. Net carbon emissions (carbon stock changes in soil, forest and products, and avoided carbon emissions due to substitution). For comparison, the reference scenario without substitution impact is shown. Underlying data used to create this figure can be found in the supporting information 4.

Compared to the reference scenario, only Scenarios 5 (textiles) and 6 (composites) resulted in a reduction in the net carbon emissions by 2056. The rest of the scenarios suggested a relatively minor impact of changing market or logging structure on the net carbon emissions of Finnish forestry at the end of the projection period. Even assuming the graphic papers to have a negative substitution impact (Scenario 7) resulted in a very similar overall substitution impact compared to the biorefinery scenario (Scenario 4). Only the scenario (8) assuming greatly diminishing displacement factors due to rapidly reducing emissions in the energy sector would lead to a significant reduction in substitution benefits and a consequent increase in net carbon emissions.

Further sensitivity analysis was carried out for both the displacement factors and the HWP half-life coefficients. The substitution impacts of the textiles scenario (5) changed ±30% in 2056, when the textile DF was increased/decreased by 50%, whereas the impacts of composites scenario (6) changed ±10% when the composite DF was changed by 50%. The impact of DF estimates is clearly large, yet even a significant change in them would not alter the main conclusions derived from the results. That is, the change of ±50% in the DFs did not change the ranking of the scenarios in terms of the net carbon emissions. When reducing the HWP half-life coefficients by 50%, the overall substitution impacts increased by 2.9-4.3% (3.9% for the reference scenario), while the net carbon emissions declined by 1.1-2.5% (1.3% for the reference scenario). That is, while end-of-life substitution benefits were increased due to an increase in the annual HWP pool outflow, the shortened life span of wood products had a greater opposite impact on net emissions. The impact of changes in half-life coefficients on results was overall small, and did not change the ranking of the scenarios.

DISCUSSION

This study examined the impact of a changing market structure in the wood-using industries on the net carbon emissions in Finland. The scenarios presented in the study are not forecasts, but they could be taken as alternative options to facilitate 'what if' analysis on the possible impact of changing market structures on net emissions and displacement factors for wood-based products.

According to the results, it is possible that the substitution benefits of wood use could be significantly increased or decreased, as compared to the current market structure. The difference in the net carbon emissions between the reference scenario and the scenario with the highest substitution impact (increased production of wood-plastic composites and textiles) in 2056 was 8.1 Mt C (29.7 Mt CO_2eq). In comparison, the difference between the net carbon emission with and without considering the substitution benefits for the reference scenario was around 9.5 Mt C (34.8 Mt CO_2eq). These values compare to total emissions (excluding the LULUCF sector) of 58.1 Mt CO_2eq in Finland in 2016.

When considering all industrial uses of wood, the average production stage displacement factor in the reference scenario was 0.33 tC/tC at the end of the simulation period (0.34 in 2016), and ranged from a minimum of 0.3 to a maximum of 0.8. Earlier studies have reported weighted displacement factors of around 0.5 tC/tC for the production stage for sawnwood, panels, and energy (Suter et al. 2017; Smyth et al. 2017).

By altering the wood use assumptions, it appears that increasing the substitution benefits is more likely to be accomplished by shifting the use of by-products from kraft pulp and heat and power production to textile, composite and wood-based panel production than by increasing sawn wood production. In addition, the results suggest that the end uses of pulp and by-products were more decisive compared to the ratio of sawn wood and pulp production. Although the scope of assumptions of this study may influence the conclusions, it seems the end-use distribution of intermediate products were more decisive for the substitution impact than the volume of intermediate-product production, given a fixed overall harvest level. Further, while the sensitivity analysis on the displacement factors showed significant variation in the results, the changes in the allocation of harvest to different end uses, i.e., the scenarios themselves, appeared to have the most influence. This said, apart from the relatively established DFs for construction, the DFs in particular for the emerging end uses of wood remain highly uncertain, and possibly not fully controlled by the sensitivity analysis carried out in this study. Moreover, there might be remarkable losses in the recovery of discarded wood products for energy use at the end-of-life and there might be similar inefficiencies in the system not captured by the assumed 5% manufacturing loss. Consequently, the overall substitution benefit may be lower than estimated.

Due to significant knowledge gaps regarding the emerging uses of wood, there is a need for balancing between expected developments in the forest products markets and the availability of data for the determination of displacement factors. For some of the emerging wood-based products, such as alternative solvent processes for regenerated cellulose fibres for textiles, the displacement factors could turn out to be higher than assumed in this study. Similarly, using lignin as a concrete admixture reduces the need for cement and water in concrete production. This may, in principle, be an effective strategy for emission reductions in the construction sector, in addition to substituting concrete for sawn wood. While such possibilities remain speculative, they do demonstrate that the range of substitution impacts depicted in this study does not necessarily represent the maximum plausible range. Our analysis also disregards the possible uptake of carbon-capture technology (e.g., BECCS) which features visibly, for example, in the IPCC 1.5 degree special report (IPCC 2018), and which, if integrated into a biorefinery, could profoundly alter our conclusions.

DFs can generally be expected to decrease over time, as all sectors of the economy are required to meet climate change mitigation obligations, which is likely to reduce fossil energy use and increase renewable energy sources and energy efficiency (Keith et al. 2015). Another highly relevant yet not well understood factor is the influence of increasing recycling rates of non-wood products on the wood-product displacement factors. In a case where a circular economy decreases the emissions of substitute products, the relative substitution benefit of wood-based products is likely to be reduced, such as in the case of replacing recycled plastic compared to primary plastic. The current study quantified only the approximate overall impact of decreasing the emissions of the energy sector. Our results show that the substitution benefit may greatly diminish, yet not completely vanish, assuming reduced emissions from the energy sector. Nonetheless, as noted by Peñaloza et al. (2018), the priority ought to be to substitute emission-intensive materials using several different approaches simultaneously to maximise the overall climate change mitigation efforts. Further research in the context of substitution and displacement factors could therefore aim to prioritise such end uses where significant emissions reductions in the existing regime are the most difficult to gain.

Besides the displacement factors, there are considerable uncertainties related to HWP half-lives, although their impact on the net carbon emissions is more marginal. In the absence of more consistent data, the half-lives applied in this study were based on widely cited IPCC reports (Pingoud et al. 2006; Skog 2003). However, as pointed out by several authors, also these values can be criticised (e.g. (Iordan et al. 2018)). While the impact of the half-life assumptions in the scenarios studied remained minor, their relative impact ought to be growing due to the general downward trend of the substitution impacts.

Irrespective of the scenario, the Finnish forestry sector was projected to remain a net carbon sink with a constant harvest level of 70 Mm^3 yr^{-1}. Further conclusion on the net carbon emissions is sensitive to initial values and the applied reference (counterfactual) scenario. In this study, different market scenarios were only contrasted against a fixed market structure, as opposed to varying harvest intensities, as the simulated harvest level already mirrored the level of industrial wood harvest in 2016. Comparing the net carbon emissions against a selected reference scenario, in which, for example, no industrial utilisation of forests takes place or in which less intensive harvests are made in the next decades, would significantly affect the conclusions drawn. In one such study (Soimakallio et al. 2016), a substitution credit of 12.4 was not sufficient to compensate for the reduction in forest carbon sink (14.6 MtC) on a 100-year time horizon. Furthermore, Seppälä et al. (2019) estimated that domestic wood harvest of 77 Mm^3 yr^{-1} instead of 57 Mm^3 yr^{-1} in Finland would mean that the average DF for increased wood use should be 2–2.4 tC/tC in timeframe 2017-2116 in order to achieve net reductions in GHG emissions. This clearly exceeds the 0.3-0.8 attained in the current study. On the other hand, the expected increase in various damage risks to forests could at least partially cancel the predicted increase in forest productivity under a changing climate (Reyer et al. 2017; Kauppi et al. 2018), which the study did not account for. That is, although the global warming has so far increased the growth of Finnish forests, natural disturbances such as storms, insect outbreaks and forest fires are estimated to substantially increase, which could again change the outcome. Further research is needed to illuminate the nature and extent of the apparent trade-off between forest carbon sinks and wood product substitution impacts, using, for example, the carbon balance indicator developed by Pingoud et al. (2016), and by considering also uncertainties related to climate change and its impacts to forests and forestry.

Finally, it is important to stress that climate policy recommendations regarding the forest sector should not be drawn based only on the results concerning the forest sink, industry structure and the displacement factor. More holistic analysis would be needed to additionally consider, for example, the long-term forest growth dynamics, the adaptation of forests to changing climate and the increasing forest disturbances, the albedo and aerosol impacts of forests, and the international carbon leakage effects (Seidl et al. 2017; Kauppi et al. 2018; Kalliokoski et al. 2019; Kallio and Solberg 2018). Thus, significant challenges remain for quantifying the overall climate impact of the industrial use of forests in a policy relevant manner, i.e., without restricting the system boundaries.

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